LCA Discussions

Eutrophication as an Impact Category

State of the Art and Research Needs

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Abstract. State of the art and research needs for the impact category eutrophication are discussed. Eutrophication is a difficult impact category because it includes emissions to both air and water – both subject to different environmental mechanisms – as well as impacts occurring in different types of terrestrial and aquatic ecosystems. The possible fate processes are complex and include transportation between different ecosystems. In some recent approaches, transportation modelling of air emissions has been included. However, in general, the used characterisation methods do not integrate fate modelling, which is a limitation. The definition of the impact indicator needs further research, too. The inclusion of other nutrients than those typically considered should also be investigated.

Keywords: Air emissions; aquatic systems; biomass production; ecosystems; eutrophication; fate modelling; impact category; nitrogen; nutrients; phosphorous; terrestrial systems; transportation; water emissions

Introduction

State of the art and research needs for Life Cycle Impact Assessment (LCIA) in general have recently been reviewed (Barnthouse et al., 1997; Finnveden and Lindfors, 1997; Udo de Haes, 1996; Udo de Haes et al., 1999). Eutrophication is generally regarded as one of the impact categories to be considered in the impact assessment. The aim of this paper is to review the state of the art and discuss research needs for the impact category Eutrophication.

On Eutrophication

It is sometimes suggested that the term "eutrophication" only refers to impacts on the aquatic systems. However, in this paper, the term refers to both impacts on aquatic and terrestrial systems. This is in line with the use of the term in many other publications (e.g. Grennfelt and Thörnelöf, 1992). Other terms are sometimes used, notably "nutrient enrichment", "nutrification" and "oxygen depletion", typically referring to the same group of impacts that are discussed here, or to some of them.

Eutrophication is a difficult impact category for several reasons. Substances that may cause the impact can be emitted to both air and water. Impacts can occur in many different types of terrestrial and aquatic systems. The fate processes are site dependent as are the impacts. The fate processes depend on different characteristics of the emitting source, environmental media, and receiving environments. The impacts depend on background loads and different sensitivities of different ecosystems. As discussed below, different nutrients may limit the growth in different ecosystems. Another complicating factor is that some impacts, e.g. increased growth, in some cases may be regarded as a positive impact rather than a negative one.

The inventory parameters that are typically assigned to "Eutrophication" are emissions of nitrogen, phosphorous and organic materials. Several methods for life-cycle impact assessment distinguish between emissions to air and water. Other nutrients which locally could be limiting, or which could be of relevance to a specific LCA case study, are typically not considered. This is further discussed below.

The concept of "limiting nutrient" is essential when discussing eutrophication. In principle, it states that, in an ecosystem, one nutrient is limiting the growth, and that there is an excess of all other nutrients. If an additional amount of the limiting nutrient is added, this will lead to increased growth. An additional amount of the other nutrients will, however, not lead to increased growth, since they are already in excess.

Terrestrial systems are often limited by nitrogen. This, for example, is the case in many ecosystems in Europe and North America (Nilsson and Grennfelt, 1988). For aquatic systems, the situation is more complex (e.g. Grennfelt and Thörnelöf, 1992). Freshwater systems are often limited by phosphorous, although there are exceptions (*ibid.*). Marine systems are often limited by nitrogen, again with exceptions. Coastal and brackish water can be limited by either phosphorous or nitrogen, or both. Other nutrients, such as silicon, can also be of importance.

The concept of "the limiting nutrient", however, is a simplification (Grennfelt and Thörnelöf, 1992) and can be misleading. Examples of complications are that the limiting nutrient may change over seasons. The limiting nutrient may also change over the years, for example as an effect of earlier loadings. The balance between different nutrients is also of importance. Since different species have different nutrient

requirements, different species may be limited by different nutrients. A change in the balance between nutrients may therefore lead to a shift in species composition. Another important aspect is the possibility of the transportation of nutrients from one ecosystem to another one. Thus, even if a nutrient is emitted to an ecosystem where it has no impact, it may be transported to another ecosystem where it can have an impact. It might therefore be that the contribution from an emission to eutrophication is always larger than zero.

An example

To illustrate the complexity and some possible fate processes, examples of the possible fate of nitrogen emitted to air will be discussed. The example is not exhaustive, since it does not describe all possible processes.

After emission to air, nitrogen can be deposited on surface water. Its subsequent fate is further discussed below. It can also be deposited on soil or vegetation. If so, it can be fixated by biomass or leached out to surface waters. The fraction being leached out in relation to the input depends on site specific aspects. In Europe it varies between approximately 5 and 80% (Grennfelt et al., 1994). If not immediately leached out, it can be taken up by growing vegetation, thus contributing to terrestrial eutrophication. The vegetation (with the nitrogen) may be exported from the ecosystem, for example, as timber or it may instead degrade. After degradation, the released nitrogen can be taken up again by growing vegetation, thus contributing to eutrophication a second time. It can, however, also be leached out to surface waters, become immobilised for some time in the soil, or be denitrified and leave the system as nitrogen gas. This nitrogen circulation can occur in every subsequent ecosystem to which it will be transported.

If the nitrogen reaches surface waters, either through direct deposition or after leaching from a terrestrial system, the surface water may be strictly phosphorous limited. In this water, the nitrogen will not contribute to eutrophication, but it can undergo further processes. It can be denitrified and thus leave the system completely, or it can be transported to another surface water that perhaps is nitrogen limited.

The nitrogen can thus be transported to nitrogen surface waters through a number of different ways, for example, direct deposition, leaching from terrestrial systems, or by transportation from other surface waters. The nitrogen may be taken up by phytoplankton, thus contributing to eutrophication. The nitrogen may in subsequent steps be exported from the system, for example together with fish. The phytoplankton may also undergo degradation, making the nitrogen available again. After this, a number of things can happen, e.g. the nitrogen can be taken up by growing phytoplankton again, it can be transported to other waters, be buried in sediments, or be denitrified and leave the system as nitrogen gas.

Overview of Suggested Methods

Around 1992-93, three methods were suggested approximately at the same time, largely independently of each other. Jensen et al. (1992) discuss eutrophication but conclude that aggregation into one impact category is difficult, because of the different types of impacts and because different nutrients are limiting in different cases. Instead they advocate an approach without any aggregation of different compounds resulting in an approach with four subcategories:

- Emissions of nitrogen to air
- emissions of nitrogen to water
- emissions of phosphorous to water
- emissions of BOD to water (Biological Oxygen Demand)

Heijungs et al. (1992) draw the opposite conclusion, suggesting a complete aggregation into one impact score. The definition of the impact indicator is **biomass production**. All emissions of nitrogen (to both air and water), phosphorous and organic material are assumed to contribute once (and only once) to the same impact. The weighting factors are based on the Redfield ratio describing the approximate stoichiometric ratio between carbon, nitrogen and phosphorous in phytoplankton.

An approach that partly can be seen as a compromise between the two earlier approaches was suggested by Finnveden et al. (1992) and further developed by Samuelsson (1993) and Lindfors et al. (1995). In this approach, terrestrial and aquatic systems are treated separately. For terrestrial systems, the impact indicator is simply the amount of nitrogen emissions to air. For aquatic systems, the definition of the impact indicator is oxygen consumption resulting from the degradation of organic material. The weighting factors are also in this case based on the Redfield ratio. It is assumed that emissions can contribute once, and only once, but sometimes may not at all contribute to the impact. Four different scenarios, or subcategories, were defined: One for the phosphorous limited case, where emissions of phosphorous and organic material are aggregated. Two for the nitrogen limited case; in one of them nitrogen emissions to water plus organic material are aggregated, and in the second also nitrogen emission to air is included in the aggregation. The separation of these two nitrogen limited scenarios is motivated because it is unknown to what extent emissions of nitrogen to air will actually reach nitrogen limited surface waters. In the final, maximum scenario, all emissions are aggregated. This last scenario is then identical to the method suggested by Heijungs et al. (1992).

After 1992, the development of methods was somewhat slowed down for a couple of years. In the discussions there were demands for site specific assessments and factors (e.g. Pujol and Boidot Forget, 1994; Potting and Blok, 1994; Blau and Seneviratne, 1995).

Hauschild and Wenzel (1996) suggested a method similar to the previous ones put forward by Heijungs et al. and Finnveden et al. They excluded organic material from the impact category and suggested an aggregation with nitrogen and phosphorous, either separately or combined.

Tolle (1997) suggested the incorporation of a regional scaling factor (1-9) increasing with present loadings. According to this approach for the U.S., large regions (e.g., the size of most states) currently receiving high nutrient loads to soils and surface water are allocated the largest potential for eutrophication. That is because already polluted regions are supposed to have already largely used the carrying capacity of the ecosystems in these regions. Additional airborne nutrient releases would then actually affect these sensitive systems.

In a recent work by Seppälä (1998), two impact categories were used for aquatic ecosystems: **Oxygen Depletion** and **Aquatic Eutrophication**. For Oxygen Depletion, emissions of BOD were used as the category indicator. For Aquatic Eutrophication, the definition of the impact indicator is increased production based on the Redfield ratio. For water emissions of nitrogen and phosphorous, site dependent transport and effect factors were determined by experts. For air emissions of nitrogen, transport factors were based on EMEP-data. It can be interesting to note that in the case study by Seppälä (1998) 6-7% of the nitrogen emitted to air was deposited in nitrogen sensitive areas. Only a small fraction of the air emissions was thus important.

For calculation of ecoprofiles intended for third party certified environmental product performance declarations, a new approach for aquatic oxygen depletion has been suggested by Lindfors et al. (1998) and Pleijel et al. (1998). Also in this case the definition of the effect is oxygen depletion, and the characterisation factors are based on the Redfield ratio. However, this approach requires a site-dependent assessment in which emissions are only considered for receiving environments, where the effect of pollutants is determined by expert judgements. No guidance is given for the situation in which a site-dependent assessment is impossible.

Characterisation factors to assess the impact from atmospheric nitrogen emission on terrestrial ecosystems are discussed and presented by Potting et al. (forthcoming). Transportation modelling is taken from the EMEP-models in a similar way as for acidification (POTTING et al., 1998). The effect is defined using a marginal approach based on the concept of critical loads. The effect is defined as the area of ecosystem that becomes unprotected as a result of that emission. This means that emissions falling on areas that are (far) below the critical load are not considered. Also emissions falling on areas which are already (far) above the critical load (i.e. they are already unprotected) are not considered. See the contribution of Potting and Hauschild (1999) for a discussion on background levels, thresholds and the different ways in which these can be taken into account in impact assessment.

Discussion and Research Issues

The SETAC-Europe working group on Life Cycle Impact Assessment discussed eutrophication (NICHOLS et al., 1996). They asked for an adaptation of earlier approaches. They concluded that several subcategories are probably necessary. A *distinction between terrestrial and aquatic systems* has already been suggested and this is probably a minimum requirement. A *further distinction between different types of aquatic systems* should also be investigated. An important aspect here is the requirement that a distinction between different ecosystems will pose on the inventory analysis.

Another issue that requires further attention is the question whether *other nutrients* should also be considered (NICHOLS et al., 1996). For example, not only nitrogen and phosphorous but also silicon are important parameters when determining the limiting factors for marine systems (GRENNFELT and THÖRNELÖF, 1992).

The *definition of the category indicators* may also need some further discussion. For aquatic systems, so far oxygen consumption or biomass production has been used. Other approaches could be worthwhile exploring. An open question is whether the definition should take into account the sensitivity of different systems. Except for the approach of Potting et al. (forthcoming), the present approaches do not integrate sensitivity.

For **terrestrial systems**, one possible, simple category indicator can be the emitted amount of nitrogen, as long as only nitrogen compounds are considered. More sophisticated indicators, taking into account differences in regional atmospheric conditions, background levels, and the difference in sensitivity of the several ecosystems in the deposition area, have been developed by Potting et al. (forthcoming) based on the concept of critical loads, still only considering nitrogen compounds. Potting et al. consider the area that becomes unprotected due to the emission (a curvilinear marginal approach). Different modifications of definitions of their indicator can be developed. Another approach is to look at the unprotected area which receives emission (i.e. the area which already receives emissions on or above the critical load) (a similar approach was used by LINDFORS et al., 1998, for acidification). Careful considerations of these, and possibly other options, should be made. For discussion, see the contribution by Potting and Hauschild (1999).

A discussion of the definition of the category indicator is also linked to more general discussion points for the whole Life Cycle Impact Assessment. One such question is whether a *marginal* or an *average approach* based on a linear or curvilinear dose/effect curve should be used. The answer on this question may depend on the goals of a specific case study (see, e.g. UDO DE HAES et al., 1999). A closely related question concerns the issue of *thresholds*. Should a threshold-approach like that of Lindfors et al (1998) and Potting et al. (forthcoming) be used, or should both below and above thresholds be considered, perhaps in separate subcategories. For discussion, see again Potting and Hauschild (1999) and also Finnveden and Potting (1999).

In the earlier approaches suggested by Jensen et al. (1992), Finnveden et al. (1992), Samuelsson et al. (1993), Heijungs et al. (1992), Hauschild and Wenzel (1996) and Tolle (1997), no fate and target modelling is included. The lack of fate and target information is a drawback of all these methods. It is still largely an open question how the modelling could be incorporated.

For **emissions to air**, both Seppälä (1998) and Potting et al. (forthcoming) use integrated assessment models for nitrogen emission. In this type of modelling, regional emission projections and modelling of fate and target systems are integrated. Background concentrations are thus included. Also target sensitivity can be included. The integrated assessment models used by both Seppälä (1998) and Potting et al. (forthcoming) also assess leaching to water systems, but they do not yet use this information in their impact modelling. This information is important for the assessment of impacts on aquatic systems from emissions to air.

For **emissions to water**, very few attempts have been made to model the fate. Both Seppälä (1998) and the IVL researchers (LINDFORS et al., 1998; PLEIJEL et al., 1998) use expert judgements for transport and fate assessment. A potentially useful research project would be to study some site specific cases carefully in order to get an understanding of the processes involved and the possible variability between different sites. Work in this direction is in process by Potting et al. (forthcoming).

Another issue for general discussion of Life Cycle Impact Assessment concerns the *spatial differentiation* (e.g. UDO DE HAES, 1996; UDO DE HAES et al., 1999). In most LCA case studies typically some, but not all, spatial information will be available. Important questions are then how to make optimal use of the spatial information that is available and the possible requirements towards the inventory analysis, if more information is wanted. A useful result would be if characterisation factors could be developed where spatially differentiated and spatially non-differentiated factors could be compatible so that the spatially differentiated factors could be used when relevant information is available.

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Call for Comments: Comments on this paper from the LCA readership are appreciated.

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